FINAL
HUMAN HEALTH RISK ASSESSMENT

KRESS CREEK/WEST BRANCH DUPAGE RIVER SITE
SEWAGE TREATMENT PLANT SITE
West Chicago Illinois

Remedial Investigation / Feasibility Study Oversight

WA No. 216-RSBD-05ZZ/Contract No. 68-W6-0025

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Acronyms and Abbreviations

µg/L	micrograms per liter
µg/m^3	micrograms per cubic meter
ANL	Argonne National Laboratory
ARAR	applicable or relevant and appropriate requirement
BEIR	NAS Biological Effects of Ionizing Radiation report
COPC	contaminants of potential concern
CSM	conceptual site model
DOE	U.S. Department of Energy
ELCR	excess lifetime cancer risk
EPC	exposure point concentrations
GI	gastrointestinal
HEAST	Health Effects Assessment Summary Tables
HHRA	human health risk assessment
HI	hazard index
HQ	hazard quotient
ICRP	International Commission on Radiological Protection
IRIS	Integrated Risk Information System
KCK	Kress Creek
MCL	Maximum Contaminated Level
mg/kg	milligrams per kilograms
mg/kg-day	Milligrams per kilogram per day
NAS	National Academy of Science
NCEA	National Center for Environmental Assessment
NCP	National Contingency Plan
NCRP	National Council on Radiation Protection and Measurements
NRC	Nuclear Regulatory Commission
ORIA	USEPA's Office of Radiation and Indoor Air
OSWER	Office of Solid Waste and Emergency Response
PCB	polychlorinated biphenyl
pCi/g	picocuries per gram
pCi/m^3	picocuries per cubic meter
PEF	particulate emission factor
PRG	Preliminary Remediation Goal

RAGS	Risk Assessment Guidance for Superfund, Part A, Volume 1
RDA	recommended daily allowance
REF	Rare Earths Facility
RESRAD	RESidual RADioactivity
RfC	Reference Concentration
RfD	Reference Dose
RI/FS	remedial investigation/feasibility study
RME	reasonable maximum exposure
SF	slope factor
STP	Sewage Treatment Plant
SVOC	semivolatile organic compounds
UCL	upper confidence limit
UNSCEAR	United Nations Scientific Committee on the Effects of Atomic Radiation
USEPA	U.S. Environmental Protection Agency
VOC	volatile organic compound
WBDR	West Branch of the DuPage River

SECTION 1

Human Health Risk Assessment

The National Contingency Plan (NCP) (Section 300.430 (d)(1)) requires that a baseline risk assessment be performed as part of a remedial investigation/feasibility study (RI/FS). The primary purpose of the baseline risk assessment is to provide risk managers with an understanding of the actual and potential risks to human health and the environment posed by a site and any uncertainties associated with the assessment. This information may be useful in determining whether a current or potential threat to human health or the environment exists that warrants remedial action (USEPA 1990; USEPA 1991).

1.1 Purpose and Objectives

The objectives of the baseline risk assessment are to:

- Develop a conceptual site model (CSM) that uses sampling and analytical data and other site information to depict the relationships between contaminant sources, migration pathways, and potentially exposed populations.
- Estimate potential exposures to human populations from contaminants in soil, sediment, surface water, fish, and groundwater at each investigation area.
- Calculate human health risks potentially associated with these estimated exposures.
- Characterize the nature of the health risks potentially associated with contaminants at the site, and discuss the degree of uncertainty associated with the baseline risk assessment.

Additional information is presented in four appendices:

- Appendix A – data and calculations used in assessing chemical risks
- Appendix B – methods and detailed results of modeling of radionuclide risks
- Appendix C – calculations used in assessing potential uptake of chemicals into fruits and vegetables
- Appendix D – evaluation of potential noncancer health risks associated with the chemical toxicity of uranium.

1.2 Risk Assessment Approach

This risk assessment was prepared in accordance with guidance provided by the U.S. Environmental Protection Agency (USEPA). The basic framework for the assessment has been derived from the USEPA Risk Assessment Guidance for Superfund, Part A, Volume 1 (RAGS) (USEPA 1989) and associated guidance documents (USEPA 1991; 1996). This risk assessment presents estimates of human health risks associated with a reasonable maximum exposure (RME) scenario. The RME estimate is considered to be a risk estimate well above the average

risk associated with contaminants at the site, but less than the maximum potential site risk. Specific factors in the RME exposure scenario included the 90 or 95 percentile values for input variables such as inhalation rate, exposure frequency and duration, and exposure concentrations based on the upper 95-percent confidence limit (UCL) of mean concentrations, or the highest concentrations detected at the site (USEPA 1989).

The RME estimate was used to provide an initial evaluation of potential human health risks associated with contaminants at each investigation area. Based on the risk estimates developed with RME assumptions, health risks may also be evaluated for selected contaminants or media using assumptions that provide more typical or reasonable estimates of potential health risks. Risk estimates using these average case exposure assumptions are not presented in the baseline risk assessment at this time.

The steps involved in preparing the baseline risk assessment are described below:

- **Data Evaluation and Selection of Contaminants of Potential Concern (COPCs):** The data evaluation is used to select the sampling and analytical data used to estimate exposure to contaminants at the site and to identify the COPCs. COPCs are those contaminants that provide the largest contribution to total site risks.

- **Exposure assessment**: An exposure assessment is conducted to estimate the magnitude of potential human exposures, the frequency and duration of these exposures, and the pathways through which humans are potentially exposed to COPCs detected at the site. Estimates of exposure are developed for the most feasible current and future land uses. The exposure assessment involves evaluating contaminant releases from the site, identifying potentially exposed populations and pathways of exposure, estimating exposure point concentrations (EPCs) for specific pathways, and estimating contaminant intake rates in humans.

- **Conceptual site model**: The CSM is used to identify and characterize potential exposure pathways and exposed populations by considering site conditions, relevant exposure scenarios, and the properties of the contaminants at the site.

- **Toxicity assessment:** Toxicity assessment involves characterizing the toxicological properties and health effects of COPCs with special emphasis on defining their dose-response relationships. From these dose-response relationships, toxicity values are derived that can be used to evaluate the potential occurrence of adverse health effects at different levels of exposure.

- **Risk characterization**: The risk characterization summarizes and combines the results of the exposure and toxicity assessments to characterize health risks, both in numerical expressions and qualitative statements. The uncertainties in the risk assessment process, and how these uncertainties influence the characterization of health risks, are discussed in this step.

At the request of the USEPA Work Assignment Manager, the output from risk assessment calculations for this report has not at this time been placed in the tabular format specified for Superfund risk assessments under RAGS Part D guidelines (USEPA 2001a).

Based on a review of the data and physical characteristics associated with the sites included in this assessment, the data were grouped into the following three investigation areas for purposes of this risk assessment:

- STP Upland - the upland portion of the West Chicago Sewage Treatment Plant (STP)
- STP River - the West Branch of the DuPage River (WBDR) from the northern edge of the STP to the confluence with Kress Creek (STP River)
- Kress Creek (KCK) - from the storm sewer outfall to the confluence with WBDR and the WBDR from the confluence to the McDowell Dam.

Both chemical and radionuclide contaminants have been detected at all three investigation areas. For the purposes of this risk assessment, the term "chemical" is generally used to refer to non-radiological contaminants and "contaminant" refers to both chemical and radiological contaminants. Health risks potentially associated with radionuclide contaminants have been characterized using RESRAD. RESRAD ("RESidual RADioactivity") was developed by the Argonne National Laboratory (ANL) for the U.S. Department of Energy (DOE). The DOE designated RESRAD for the evaluation of radioactively contaminated sites. The U.S. Nuclear Regulatory Commission (NRC) has approved the use of RESRAD for dose evaluation by licensees involved in decommissioning, NRC staff evaluation of waste disposal requests, and dose evaluation of sites being reviewed by NRC staff. Finally, the USEPA Science Advisory Board reviewed the RESRAD model, and the USEPA used RESRAD in its rulemaking on radiation site cleanup regulations.

1.3 Data Evaluation and Selection of Contaminants of Potential Concern

This section describes the steps taken to organize the sampling and analytical data into a form appropriate for a baseline risk assessment. These steps are intended to ensure that appropriate and reliable data are carried through the quantitative steps in the risk assessment. This evaluation discusses the sources of sampling and analytical data, the selection of data for estimating EPCs, and the selection of COPCs. EPCs were estimated for COPCs in order to quantify health risks associated with contaminants detected at each investigation area. Estimation of EPCs is discussed in Section 1.4.9.

1.3.1 Data Sources

Section 3 of the Kerr-McGee RI report for the Kress Creek and STP Sites (BBL 2004) details the field efforts associated with soil, sediment, surface water, fish, and groundwater samples used for this risk assessment.

1.3.2 Selection of Contaminants of Potential Concern

The selection of COPCs for the human health risk assessment (HHRA) is a step-wise process that evaluates the analytical data associated with each exposure area. Contaminants that pass the selection process are retained for further analysis in the risk assessment. The COPCs selection process typically results in a reduction in the number of contaminants carried through to the risk assessment compared to the total number of contaminants

identified during site characterization. Its purpose is to allow the risk assessment to focus on the contaminants that are site-related and likely to pose a significant risk.

1.3.2.1 Comparison of Site Data with Background Concentrations

Site-specific background concentrations were established using numerous samples collected from locations within the boundaries of each investigation area that have not been affected by site activities. Background sediment, surface water, and fish tissue samples were collected from locations within the boundaries of the STP River and the KCK study areas.

A contaminant was removed as a COPC if the maximum concentration of the contaminant from site samples was not detected at a concentration above the site-specific maximum detected background concentration.

1.3.2.2 Frequency of Detection

Contaminants detected at a frequency of less than 5 percent are removed as COPCs, particularly if site historical information indicates that such contaminants were unlikely to have been associated with site activities. In such cases, contaminants present in a particular medium at a frequency of less than 5 percent (for sites where there were 20 or more samples collected from that medium) were excluded as COPCs.

1.3.2.3 Comparison of Site Data to Screening Values

The purpose of this comparison is to remove contaminants that are not likely to significantly contribute to overall risk. The maximum detected concentrations of chemicals detected in onsite soil and sediment/floodplain soil were compared to the USEPA Region IX Preliminary Remediation Goals (PRGs) for residential soil (USEPA 2002a). The maximum detected concentrations of chemicals detected in groundwater were compared to USEPA Region IX PRG's for tap water and Maximum Contaminated Levels (MCLs) established by the USEPA's Office of Water, Office of Science and Technology (USEPA 2002c). The maximum detected concentrations of chemicals detected in surface water were compared to National Recommended Water Quality Criteria (USEPA 2002b) for consumption of organisms and water. Risk-based screening values are not available for fish tissue; therefore, comparison of site data to screening values was not conducted for fish data. Similarly, risk based screening values are not available for radionuclides so comparison of site data to screening values was not performed for radionuclides.

1.3.2.4 Exclusion of Essential Nutrients

Essential nutrients are those chemicals considered essential for human nutrition. Recommended daily allowances (RDAs) are developed for essential nutrients to estimate safe and adequate daily dietary intakes (National Academy of Science [NAS] 1989). The following metals are considered to be essential nutrients and were excluded as COPCs: calcium, magnesium, potassium, and sodium.

1.3.2.5 Naturally Occurring Radionuclides and Daughter Products

Naturally occurring radionuclides that were unlikely to have been associated with contaminant releases to the site (e.g., K-40) were excluded as COPCs. Radionuclide daughter products that were part of the decay chains for radium or uranium were reported as analytical

results in soil or sediment samples. However, for purposes of assessing risks, the contribution to total risks from daughter products was calculated within RESRAD. Therefore, while daughter product risks were addressed as part of the risk assessment, the analytical results in soil or sediment for daughter products were not used directly in the risk assessment.

1.4 Exposure Assessment

Exposure assessment is the estimation of the magnitude, frequency, duration, and routes of exposure to chemicals or radionuclides. Human exposure to contaminants is typically evaluated by estimating the amount of a substance that could come into contact with the lungs, gastrointestinal (GI) tract, or skin during a specified period of time. This exposure assessment is based on scenarios that define human populations potentially exposed to COPCs that may originate from soil, sediment, surface water, and groundwater at the site. The potential pathways of exposure, frequency and duration of potential exposures, rates of contact with air and soil, and the concentrations of contaminants in air, soil, surface water, or groundwater are considered in the exposure assessment. Contaminant intakes and associated risks have been quantified for all exposure pathways considered potentially complete. This section describes the assumptions, data, and methods used to evaluate the potential for human exposure to COPCs originating from the study sites. The exposure assessment involves the following steps:

- Identification of potential exposure pathways
- Evaluation of the environmental fate and transport of contaminants
- Identification of potentially exposed populations
- Selection of exposure scenarios
- Estimation of EPCs used to quantify contaminant intakes
- Quantification of contaminant intakes for each exposure pathway

The information developed through these steps was then used to develop exposure scenarios. An exposure scenario considers the sources of the contaminants that could come into contact with the subject population. Exposure prediction models based on the fate and transport of the contaminants are then used to evaluate the pathways from the sources to subject population.

The results from the exposure assessment are estimates of contaminant intakes through each exposure pathway to members of each potentially exposed population. The results from the exposure assessment are then combined with the results from the toxicity assessment to characterize health risks associated with contaminants in each investigation medium (e.g., soil, sediment, surface water, fish, and groundwater) at each investigation area.

The basis for estimating exposure to contaminants in this baseline risk assessment is the RME scenario. The RME is defined as the highest exposure that is reasonably expected to occur at a site. The intent of the RME is to develop a conservative estimate of exposure (i.e., well above the average case) that is still within the range of possible exposures. The USEPA states that " . . . a determination of 'reasonable' cannot be based solely on quantitative information, but also requires use of professional judgement" (USEPA, 1989). Specifically, the RME scenario includes exposure concentrations based on the 95-percent UCL of mean concentrations, or the highest concentrations, detected at a site. Default exposure factors

corresponding to RME for different exposure scenarios have been developed by the USEPA (USEPA 1991). Exposure factors developed in RESRAD for radionuclides are intended to be consistent with an RME scenario.

1.4.1 Conceptual Site Model

The CSM is a three-dimensional "picture" of site conditions that illustrates contaminant distributions, release mechanisms, exposure pathways and migration routes, and potentially exposed populations (USEPA 1996). The CSM documents current site conditions, and uses site information to illustrate exposure pathways that may arise through contaminant release and migration to human populations. The CSM is used to organize site data according to contaminant source characteristics, migration pathways, affected media, and contaminant properties. The CSM is presented in Figures 1-1a through 1-1c.

1.4.2 Site Description

This subsection briefly describes the characteristics of each investigation area. The locations of each investigation area are shown on Figure 1-2, and additional descriptions of each can be found in Sections 1 and 2 of the Kerr-McGee RI report for the Kress Creek and STP Sites (BBL 2004).

1.4.2.1 Sewage Treatment Plant Upland

The STP Upland area includes the West Chicago Sewage Treatment Plant property, which is owned and operated by the City of West Chicago (located at Illinois Routes 59 and 38, Sarana Drive in West Chicago).

The STP is situated in a low-density development area with residential areas to the west, scattered residences to the east and south, the Blackwell Forest Preserve to the south and West DuPage Woods Forest Preserve to the north. The closest residence is located downwind, about 300 feet to the east. The residential area west of the STP is more than 500 feet from the site.

1.4.2.2 Sewage Treatment Plant River

The STP River area consists of the West Branch DuPage River from the northern boundary of the STP property to the confluence of the West Branch and Kress Creek.

Residences along the river are primarily single family homes in unincorporated West Chicago. Many of the homes in the Edgewood Walk subdivision are located along the WBDR, south of the KCK confluence. Many of these homes lie within the 100-year floodplain. Much of the area surrounding the WBDR is forest preserve land. Wetland areas are connected to the water bodies mostly during high flows. The river has gravel banks and a stony streambed covered with vegetation.

1.4.2.3 Kress Creek

The KCK Site includes a 1.2 mile stretch of KCK from the storm sewer outfall to the confluence with the WBDR and the WBDR from the confluence approximately 5.2 miles downstream to the McDowell Dam. KCK is a surface stream located south of the decommissioned Kerr-McGee Rare Earths Facility (REF) and has received radiologically contaminated wastes via a storm sewer discharge from the Kerr-McGee REF. It flows about

330 feet through undeveloped property, through a developed subdivision where it expands into Gunness Lake (Figure 1-2). The creek flows through open fields, a West Chicago park, beneath Wilson Street Bridge, and through the grounds of a religious temple, where it reaches Illinois Route 59. The creek continues its course beneath Illinois Route 59 and behind the Edgewood Walk Subdivision before reaching its confluence with the WBDR. From the confluence, the WBDR flows through residential areas and through forest preserves to the Warrenville Dam, then through commercial areas, residential areas and a forest preserve to the McDowell Dam.

The KCK study area contains a diversity of land use, ranging from high-density residential to relatively mature deciduous woodland. Residences along KCK and the WBDR in unincorporated West Chicago are primarily single-family homes in subdivisions. Many of the homes in the May and Joy Street Subdivision are located along KCK. Many of these homes lie within the 100-year floodplain.

1.4.3 Potential Receptor Populations

1.4.3.1 STP Upland

Currently exposed populations at the STP Upland exposure area potentially include maintenance workers and construction workers. Maintenance workers perform general maintenance work around the STP facility, including some limited intrusive work in subsurface soil. Construction workers perform intermittent construction work, including work in subsurface soil. Reasonably anticipated uses of the STP area potentially include the construction of residential developments on the facility area. Potential future exposed populations include residents of new homes and construction workers.

1.4.3.2 STP River

Currently exposed populations at the STP River exposure area potentially include residents and visitors that use the WBDR for recreational purposes such as canoeing, fishing or swimming. Future land use and potentially exposed populations are anticipated to remain the same. Reasonably anticipated use of areas potentially include the construction of more residential developments along the river. Potential future exposed populations include residents of new homes.

1.4.3.3 KCK

Currently exposed populations at the KCK exposure area potentially include residents and visitors that use KCK and WBDR for recreational purposes such as fishing or swimming. Future land uses and potentially exposed populations are anticipated to remain the same. Reasonably anticipated use of areas potentially include the construction of more residential developments along the creek and river. Potential future exposed populations include residents of new homes.

1.4.4 Identification of Potential Exposure Pathways

An exposure pathway describes the mechanism through which a contaminant comes into contact with a receptor (i.e., potentially exposed individual). There must be a complete exposure pathway from the source of contaminants in the environment (in soil or groundwater) to human receptors in order for contaminant intake to occur. In this section,

the complete exposure pathways are chosen from all potential pathways, and are further evaluated.

A complete contaminant exposure pathway consists of the following elements:

- A source of contaminant release to the environment (i.e., contaminant concentrations in soil)
- A transport medium (i.e., soil, groundwater, or air).
- An environmental fate and transport mechanism (i.e., migration through soil to groundwater; volatilization into air).
- A point of contact (known as the exposure point) for receptors with the COPCs.
- A route of intake for the contaminant into the receptor (ingestion, inhalation, or dermal absorption).

If one of these elements is missing, then the exposure pathway is incomplete and there is no intake (or subsequent health risk) associated with that pathway. The presence or absence of any of these elements depends on the specific conditions found at the site.

The exposure scenarios associated with each investigation area are summarized in Table 1-1. Potentially exposed populations, media, and potential routes of exposure are also presented in Table 1-1 for COPCs identified at each investigation area.

1.4.5 Description of Contaminant Sources

The waste materials transported from the REF contained a wide range of constituents, including tailings from processed ores, possibly treated ores, and waste products from other process and manufacturing activities. Numerous sampling and analysis programs were conducted on the original waste materials at the REF. The radiological residuals include thorium, uranium, and their radioactive decay products. A discussion of contaminant sources is presented in the Kerr-McGee RI report for the Kress Creek and STP Sites (BBL 2004).

1.4.6 Evaluation of Environmental Fate and Transport

The environmental fate and transport evaluation was performed to estimate EPCs in media where sampling and analytical data were not available. Samples were available to evaluate potential direct contact exposures (ingestion and dermal contact). However, analytical data (air sampling data) were not available to estimate potential inhalation exposures. Therefore, emissions of contaminants from soil, sediment, and groundwater to air were estimated using modeling to provide EPCs in air.

The potential air exposure pathway includes the inhalation of contaminants adhering to soil particulates that become suspended into the air. Concentrations of radionuclides in air were estimated using RESRAD. Concentrations of particulate-bound chemicals in air were estimated using a particulate emissions factor (PEF), as shown in Appendix A (USEPA, 1996). VOCs were not detected in any soil or sediment samples collected from each investigation site; therefore, the inhalation of VOCs to ambient air was not evaluated. Soil

particulates suspended into the air through wind entrainment could potentially produce concentrations in air, so that contaminant intake potentially could occur through inhalation.

Section 5 of the Kerr-McGee RI report for the Kress Creek and STP Sites (BBL 2004) provides an evaluation of the fate and transport of the site related contaminants.

1.4.7 Potential Release Mechanisms and Exposure Pathways

The release mechanisms for contaminants detected in surface soil include the following:

- Suspension of dust-containing contaminants into the air
- Emissions of radon to the air from radionuclide decay in soil
- Infiltration and/or percolation of contaminants into groundwater
- Direct contact with contaminants (potentially resulting in soil ingestion or dermal exposure[1])
- Infiltration and/or percolation of contaminants into groundwater discharging to surface water
- Surface runoff of contaminants into surface water or sediments,
- Accumulation of contaminants into fish tissue

The primary release mechanisms for contaminants detected in subsurface soil include the following:

- Suspension of dust-containing contaminants into the air
- Emissions of radon to the air from radionuclide decay in soil
- Direct contact with contaminants (potentially resulting in soil ingestion or dermal exposure)

Exposure pathways, exposure routes, and potential exposure points are depicted in the CSM presented in Figures 1-1a through 1-1c and are summarized in Table 1-1.

1.4.8 Selection of Exposure Pathways to be Quantified

1.4.8.1 Exposure to Contaminants in Soils

Future residents and current or future maintenance and construction workers could come into contact with contaminants in surface soils at the STP Upland. Exposure to surface soils at the STP Upland assumes there is open access to waste material and the area is not enclosed by a fence. In general, exposure to surface soils could occur via inadvertent ingestion, skin contact, or by inhaling dusts. Exposure to radionuclides can also occur through external gamma radiation. The dermal exposure pathway does not include radiological compounds adhering to skin or skin dose from beta emissions from soil. It is anticipated that risks potentially associated with this pathway are a small contribution to total site risks, and this pathway is not included in the RESRAD model.

[1] Note that risks from dermal exposure to radionuclides is not included in the risk assessment, and that this exposure pathway is not included in the RESRAD model. It is anticipated that risks from dermal exposure to radionuclides in soil are a small contribution to total site risks.

1.4.8.2 Exposure to Contaminants in Groundwater

Contaminants have been detected in the shallow groundwater beneath the STP Upland. Future construction workers could come into contact with contaminants in groundwater. Construction workers may excavate soils for utility installation, maintenance, or other purposes. Exposure to groundwater could occur via dermal contact with contaminants during these activities.

There is no indication that shallow groundwater at the facility will be used for drinking water supply. Drinking water is supplied by groundwater wells that are screened in deeper groundwater sources, not the shallow groundwater. Therefore, it is unlikely that maintenance workers or nearby residents would be exposed to shallow groundwater through ingestion as drinking water, dermal contact, or inhalation of contaminants during regular household activities.

1.4.8.3 Exposure to Contaminants in Sediment/Floodplain Soil

Future residents or recreational users could come into contact with contaminants in sediment/floodplain soils. In general, exposure to surface sediment/soil could occur via inadvertent ingestion, skin contact, or by inhaling dusts. Exposure to radionuclides can also occur through external gamma radiation. The dermal exposure pathway does not include radiological compounds adhering to skin, or skin dose from beta emissions from soil. It is anticipated that risks potentially associated with this pathway are a small contribution to total site risks, and this pathway is not included in the RESRAD model.

1.4.8.4 Exposure to Contaminants in Surface Water

Human exposure to contaminants in surface water could occur through incidental ingestion or dermal contact during swimming or wading. The dermal exposure pathway does not include radiological compounds adhering to skin, or skin dose from beta emissions from sediment. In addition, exposure from beta emissions while submerged in surface water is not included as a route of exposure. It is anticipated that risks potentially associated with this pathway are a small contribution to total site risks.

Contaminants have been detected in surface water at the STP River and the KCK site. Currently, contact with surface water could occur through recreational activities (swimming or wading) by residents living in the areas near the STP River or the KCK site or by visitors to these areas.

1.4.8.5 Exposure to Contaminants in Fish Tissue

Human exposure to contaminants could occur through the ingestion of fish caught during recreational fishing in contaminated surface water. Contaminants have been detected in fish collected from the STP River or the KCK site. Since residents do live nearby and have access to these areas, exposure to contaminants through ingestion of fish could occur.

1.4.8.6 Exposure to Contaminants in Fruits and Vegetables

Human exposure to contaminants could occur through ingestion of fruits and vegetables from gardens raised in contaminated soils. Potential contribution to total risks from radionuclides through the fruit and vegetable ingestion pathway have been evaluated in RESRAD (see Appendix B).

Potential contribution to total risks from chemicals through fruit and vegetable ingestion were evaluated using the equations and assumption presented in *Human Health Risk Assessment Protocol for Hazardous Waste Combustion Facilities, Peer Review Draft* (USEPA 1998). These equations are presented in Appendix C.

1.4.9 Estimation of Exposure Point Concentrations

RME concentrations were used to estimate potential contaminant intakes. RME concentrations can be estimated by direct measurement at a point of contact or by modeling a contaminant release from a source and transport to the exposure point. This assessment uses concentrations at the point of assumed contact to estimate RME concentrations. RME concentrations are assumed to remain constant for the duration of exposure. Physical, chemical, or biological processes that could reduce contaminant concentrations over time are not factored into the estimate of RME concentrations. This conservative assumption may overestimate exposure.

RME concentrations for contaminants quantitatively evaluated in the risk assessment were calculated using the less of either the maximum concentration or the 95-percent UCL on the mean concentration in media of concern.

The 95-percent UCL of the mean was calculated for a media in which ten or more samples were collected. The maximum detected concentration was used in place of the 95-percent UCL when the calculated 95-percent UCL was greater than the maximum detected value. The Shapiro-Wilks W-test using an alpha value of 5 percent was used to determine if the data fit a lognormal or normal distribution. If the W-test was inconclusive, the maximum concentration was used to calculate total site risks. For small data sets (i.e., fewer than ten samples), the maximum detected concentration was used to calculate total site risks.

The 95-percent UCL for a lognormal distribution was calculated as follows:

$$95\%\,UCL = \exp\left(TM + 0.5\,s^2 + \frac{s\,H}{\sqrt{n-1}} \right)$$

where:

exp	=	natural log
TM	=	log-transformed mean
s	=	standard deviation of the transformed data
H	=	H-statistic (e.g., from Table A12 published in Gilbert 1987)
n	=	sample size

The 95-percent UCL for a normal distribution was calculated as follows:

$$95\%\,UCL = NM + \frac{t\,s}{\sqrt{n}}$$

where:

NM	=	normal arithmetic mean
t	=	t-statistic (e.g., from Table A2 published in Gilbert 1987)
s	=	standard deviation
n	=	sample size

1.4.10 Quantification of Exposure

This section describes the methods for calculating potential contaminant intakes for the populations and exposure pathways selected for quantitative evaluation.

For chemicals the intakes calculated in this section are expressed as the amount of chemical at the exchange boundary (skin, lungs, or gut) and available for absorption. Estimates of chemical intakes are based on RME assumptions. Chemical intakes were estimated for populations and exposure pathways identified under current and future land uses. The following subsection describes the assumptions and calculations used to estimate the chemical intake for each exposure pathway, and presents the intake rates associated with each exposure pathway. The equations and assumptions used to estimate chemical intakes are presented in Appendix A.

Evaluation of potential radionuclide intakes and risks were evaluated using the RESRAD model. Further description of the methods for quantifying exposures and risks using RESRAD are presented in Appendix B.

1.4.10.1 Standard Exposure Factors

Intake equations require variables (exposure factors) specific to particular exposure scenarios. Exposure factors often are assumed values and their magnitude influences the estimates of potential exposure. The reliability of selected values also contributes to uncertainty in the resulting estimates. Many exposure parameters used in this assessment are default values provided by standard USEPA guidance. The intake parameters and equations used to evaluate chemical and radionuclide exposure for each receptor are summarized in Tables 1-2 through 1-10.

The methods presented in USEPA's RAGS, Part E (USEPA 2001) for estimating dermal exposure to water were used to evaluate dermal exposure to surface water during swimming and wading for residents and dermal exposure to groundwater in an open excavation for construction workers. The non-steady state model or pseudo steady-state model was used to estimate the dermally absorbed dose per event for organic constituents (USEPA 2001). If the exposure time (or event time, t_{event}) was shorter than the time to reach steady state (t*), the non-steady state model was used. If t_{event} was greater than t*, the pseudo-steady state model was used. For inorganics, the absorbed dose was calculated using a steady-state approach. These equations used for models are shown in Appendix A.

For residential exposure to chemicals in soil, sediment, surface water, and fish, lifetime age-adjusted intakes were calculated to evaluate carcinogenic risk. Age-adjusted exposure factors were calculated using the equations presented in Tables 1-2, 1-4, and 1-5. For the evaluation of noncarcinogenic risks for residents, child exposure factors were used to estimate chemical intakes through soil, sediment, surface water, and fish.

1.5 Toxicity Assessment

The toxicity assessment describes the relationship between the magnitude of exposure to a contaminant and adverse health effects. This assessment provides, where possible, a

numerical estimate of the increased likelihood and/or severity of adverse effects associated with contaminant exposure (USEPA 1989). This section provides a brief description of the adverse effects and toxicity values used to characterize health risks for the contaminants of potential concern detected at the site.

For purposes of the toxicity assessment, the COPCs have been classified into two broad categories: noncarcinogens and carcinogens. This classification has been selected because health risks are calculated quite differently for carcinogenic and noncarcinogenic effects, and separate toxicity values have been developed for carcinogenic and noncarcinogenic effects. These toxicity values represent the potential magnitude of adverse health effects associated with exposure to contaminants, and are developed by the USEPA. Toxicity studies with laboratory animals or epidemiological studies of human populations provide the data used to develop these toxicity values. These values represent allowable levels of exposure based upon the results of toxicity studies or epidemiological studies. The toxicity values are then combined with the exposure estimates (presented in the previous section) in the risk characterization process to estimate the potential for adverse health effects from contaminants at the site.

Toxicity values (cancer slope factors [SFs] and Reference Doses [RfDs]) used in the risk assessment were obtained from these sources:

- The Integrated Risk Information System (IRIS), a database available through by the USEPA National Center for Environmental Assessment (NCEA) in Cincinnati, Ohio. IRIS, prepared and maintained by the USEPA, is an electronic database containing health risk and USEPA regulatory information on specific chemicals (USEPA 2003).
- The Health Effects Assessment Summary Tables (HEAST), provided by the USEPA Office of Solid Waste and Emergency Response (OSWER) (USEPA 1997). HEAST is a compilation of toxicity values published in various health effects documents issued by the USEPA.

The toxicity values used to characterize health risks potentially associated with chemicals at the site are presented in Table 1-11. Further descriptions of these values are presented below. The toxicity values for radionuclides are discussed n Section 1.5.4.

1.5.1 Noncarcinogenic Effects (Chemicals and Uranium)

Noncarcinogenic effects were evaluated using either RfDs or Reference Concentrations (RfCs), developed by the USEPA. The RfD is a health-based criterion, expressed as chemical intake rate in units of milligrams per kilogram per day (mg/kg-day), used in evaluating noncarcinogenic effects. The RfD is based on the assumption that thresholds exist for certain toxic effects such as liver or kidney damage, but may not exist for other toxic effects such as carcinogenicity. In general, the RfD is an estimate (with uncertainty spanning perhaps an order of magnitude) of a daily exposure to the human population (including sensitive subgroups) that is likely to be without an appreciable risk of deleterious effects during a lifetime of exposure (USEPA 1989). RfDs are developed for oral routes of exposure. The RfC, expressed as a concentration in air with units of mg/m^3, is used to evaluate adverse effects from inhalation exposure. Exposures to chemical substances in air were estimated in units of mg/kg-day. Therefore, RfCs were converted to inhalation RfDs, as follows:

$$RfD_i = RfC\,(mg/m^3) \times \left(\frac{20\ m^3/day}{70\ kg}\right)$$

where RfD_i is the inhalation RfD, 20 m^3/day is the daily inhalation rate, and 70 kg is body weight.

Potential health risks associated with exposure to noncarcinogenic compounds were evaluated by calculating a Hazard Quotient (HQ). The potential HQ was calculated as the ratio of the intake to the RfD, as follows:

$$HQ = \frac{Intake}{RfD}$$

If the estimated daily intake for any single chemical is greater than its RfD, the HQ will exceed unity. A HQ that exceeds unity indicates that there is a potential for adverse health effects associated with exposure to that chemical.

A Hazard Index (HI) is calculated to assess the potential for noncarcinogenic effects posed by more than one chemical. The HI approach assumes that simultaneous subthreshold exposures to several chemicals could result in an adverse health effect. It also assumes that the magnitude of the adverse effect will be proportional to the sum of the ratios of the subthreshold exposures to the acceptable exposure (the RfD). The HI is equal to the sum of the HQs, and is calculated as follows:

$$HI = \frac{E_1}{RfD_1} + \frac{E_2}{RfD_2} + \cdot\ \cdot\ \cdot + \frac{E_i}{RfD_i}$$

where Ei is the exposure level (or intake) for the ith chemical, and RfDi is the RfD for the ith chemical. E and RfD are expressed in the same units (mg/kg-day), and represent the same exposure period (i.e., chronic, subchronic, or short term). A further evaluation of hazard was performed by calculating HI for groups of chemicals with the same critical toxic effect or that cause toxicity in the same target organ, when necessary. RfD values and target organ used in calculating HQs are presented in Table 1-11.

Uranium, a radionuclide, can be associated with noncarcinogenic toxic effects such as kidney damage. The noncancer risks from uranium were evaluated separately in Appendix D.

1.5.2 Carcinogenic Effects (Chemicals)

Evidence of carcinogenicity of a contaminant comes from two sources: lifetime studies with laboratory animals and human studies where excess cancer risk is associated with exposure to the contaminant. Unless evidence to the contrary exists, if a carcinogenic response occurs at the exposure levels studied (typically high doses), it is assumed that responses will occur at all lower doses. Exposure to any level of a carcinogen is then considered to have a finite risk of inducing cancer.

Because risks at low levels of exposure cannot be quantified directly by either animal or epidemiological studies, mathematical models are used to extrapolate from high to low doses. The linearized multi-stage model for low dose extrapolation is recommended by regulatory agencies (USEPA 1986). Use of the linearized multi-stage model leads to a conservative upper

bound estimate of risk. The linearized multi-stage model incorporates a procedure for estimating the largest possible slope at low doses that is consistent with experimental dose-response data (use of a large slope tends to produce a higher estimate of cancer risk). The most sensitive species of animal is used for extrapolation to humans (i.e., the assumption being that man is as sensitive as the most sensitive animal species). The true risk to humans is not likely to be higher than the estimate and is most likely lower, and could even be zero.

Numerical estimates of cancer potency are presented as slope factors (SFs). Under an assumption of dose-response linearity at low doses, the SF defines the cancer risk from a continuous lifetime exposure to one unit of carcinogen (in units of risk per [mg/kg-day]). For chemicals, individual cancer risk was calculated as the product of exposure to a chemical in (mg/kg-day) and the SF for that chemical in $(mg/kg\text{-}day)^{-1}$, as follows:

$$Risk = Intake \times SF$$

Cancer risks from exposure to multiple carcinogens and multiple pathways were assumed to be additive, based on the USEPA carcinogen risk assessment guidelines (USEPA 1996). The theoretical probability of a single receptor person developing cancer as a consequence of exposure to two or more chemicals and by two or more exposure pathways was calculated by summing the risk estimates for each receptor for each chemical in the appropriate scenarios using the following equation:

$$Total\ ELCR = (I_1 \times SF_1) + (I_2 \times SF_2) + ... + (I_i \times SF_i)$$

where I is the exposure level (or intake) (mg/kg-day) for the ith chemical, and SF is slope factor $(mg/kg\text{-}day)^{-1}$ for the ith chemical (mg/kg-day).

Each SF is accompanied by a weight-of-evidence classification. The weight-of-evidence classification considers the available data for a chemical to evaluate the likelihood that the chemical is a potential human carcinogen. The evidence is characterized separately for studies in humans and studies in laboratory animals as sufficient, limited, inadequate, no data, or evidence of noncarcinogenicity. The USEPA recommends that cancer risk estimates should always be accompanied by a weight-of-evidence classification to indicate the strength of evidence that a chemical is a human carcinogen (USEPA 1986; USEPA 1989). The SFs and weight-of-evidence classifications for each chemical are summarized in Table 1-11. A description of the weight-of-evidence classification is also presented in Table 1-11.

Potential carcinogenic effects associated with radionuclides are discussed further in Section 1.5.4 and Appendix B.

1.5.3 Adjustment of Oral Toxicity Values for Dermal Exposure

Health risks from dermal exposure to chemicals were estimated by modifying oral RfDs for noncarcinogenic effects or SFs for carcinogenic effects by chemical-specific GI absorption factors. The oral administered doses are transformed into absorbed doses for comparison of intakes for dermal exposure. Administered doses for chemicals with noncarcinogenic effects were modified to absorbed doses by dividing the oral intake by the GI absorption factor and the RfD. Administered doses for chemicals with carcinogenic effects were modified to absorbed doses by multiplying intake by the GI absorption factor and the SF.

1.5.4 Radionuclides

The USEPA classifies all radionuclides as known human carcinogens, based on their property of emitting ionizing radiation and on the extensive weight of evidence provided by epidemiological studies of radiogenic cancers in humans. With the exception of uranium radioisotopes, evaluation of the health risks consider the carcinogenic effects of radionuclides only. In most cases, cancer risks are limiting, exceeding both mutagenic and teratogenic risks (USEPA 1997). The noncarcinogenic risks from uranium were evaluated separately in Appendix D.

Excess lifetime cancer risks from intake of radionuclides were estimated using cancer slope factors (or risk coefficients) presented in USEPA's *Cancer Risk Coefficients for Environmental Exposure to Radionuclides* (USEPA, 1999). These values also have been incorporated into the updated Health Effects Assessment Summary Tables (HEAST) for radionuclides (USEPA, 2004). These values also are incorporated into RESRAD, as discussed in Appendix B. The cancer slope factors provide an estimate of the risk to an average individual exposed to radionuclides over a lifetime, based on a site-specific exposure scenario. Selected radionuclides and radioactive decay chain products are designated with the suffix "+D" (e.g., U-238+D, Ra-226+D) to indicate that cancer risk estimates for these radionuclides include the contributions from their short-lived decay products, assuming equal activity concentrations (i.e., secular equilibrium) with the principal or parent nuclide in the environment.

1.6 Risk Characterization

Risk characterization involves estimating the magnitude of the potential adverse health effects under study. This is accomplished by combining the results of the dose-response and exposure assessments to provide numerical estimates of potential health effects. These values represent comparisons of exposure levels with appropriate RfDs and estimates of excess cancer risk. Risk characterization also considers the nature of and weight of evidence supporting these estimates, as well as the magnitude of uncertainty surrounding such estimates.

A summary of the numerical risk estimates is presented in site-specific risk evaluation sections for each investigation area. Section 2 presents the results for STP Upland, Section 3 presents the results for STP River and Section 4 presents the results for Kress Creek. Generally, the USEPA considers action to be warranted at a site to reduce risks when ELCRs exceed 1×10^{-4}. The need for remedial action for risks falling between 1×10^{-4} to 1×10^{-6} is judged on a case-by-case basis. Risks less than 1×10^{-6} generally are of no concern to regulatory agencies. A noncancer HI (the ratio of the contaminant intake to the RfD) greater than 1.0 indicates some potential for adverse noncancer health effects associated with exposure to contaminants of potential concern (USEPA 1991b).

Although the risk assessment produces numerical estimates of risk, these numbers do not predict actual health outcomes. The estimates are calculated to overestimate risk, and thus any actual risks are likely to be lower than these estimates, and may even be zero.

1.7 Uncertainties

The numerical estimates of health risks must be accompanied by a discussion of the uncertainties inherent in the assumptions used in estimating these risks. Uncertainties in the risk estimation process may result in the numerical estimates either understating or overstating health risks associated with contaminants in the sites. Some level of uncertainty is introduced each time an assumption is made. The aggregation of several assumptions also magnifies the uncertainties in the numerical risk estimates.

Sources of uncertainties in the exposure assessment include assumptions as to what constituted complete exposure pathways based on the site setting; applicability of the models, data interpretations, and sampling and analytical data used to estimate EPCs; and the transport, fate, and intake parameters used to estimate exposure concentrations and intake rates. Most often, the sampling and analytical data and the toxicity assessment provide the largest contributions to overall uncertainty in the risk assessment. Sources of uncertainties in the toxicity assessment include those associated with extrapolation of adverse effects from laboratory animals to humans and from high levels of exposure to low levels encountered in the environment.

SECTION 2

Summary of HHRA Results for STP Upland

This section presents a summary of HHRA results conducted on STP Upland using methodologies presented in Section 1.

2.1 Data Evaluation and Contaminants of Potential Concern

This risk assessment was performed for STP Upland using analytical data obtained during the RI/FS in 1993 and supplemental data collection in 1994. The list of samples used for HHRA is presented in Table 2-1 and detailed description of samples and data collection activities can be found in Section 3 of the Kerr-McGee RI Report for the Kress Creek and STP Sites (BBL 2004).

2.1.1 Onsite Soil

One hundred sixty-nine samples (0- to 2-foot interval) were collected within the STP site boundary and were analyzed for metals, VOCs, semivolatile organic compounds (SVOCs), and radionuclides. Table 2-2 summarizes analytical results of detected contaminants in STP Upland soil. Four metals (arsenic, iron, lead, and manganese) and three SVOCs (benzo(a)pyrene, benzo(b)fluoranthene, and dibenz(a,h)anthracene) were detected at concentrations higher than screening levels in soil and were identified as COPCs. Additionally, five radionuclides (radium-226, radium-228, thorium-232, uranium-235, and uranium-238) were identified as COPCs for soil. Table 2-3 presents a summary of COPC selection process for STP Upland soil.

2.1.2 Groundwater

Thirteen groundwater samples were collected within the STP Upland site boundary and were analyzed for metals, VOCs, SVOCs, and radionuclides. Table 2-4 summarizes analytical results of detected contaminants in groundwater. Three metals (arsenic, lead, and vanadium) and three VOCs (bromodichloromethane, chloroform, and dibromochloromethane) were detected at concentrations higher than screening levels in groundwater and were identified as COPCs. Additionally, five radionuclides (radium-226, radium-228, thorium-232, uranium-235, and uranium-238) were identified as COPCs for groundwater. Table 2-5 presents a summary of COPC selection process for STP Upland groundwater.

2.2 Potential Exposure Pathways

Potential exposure pathways from onsite soil and groundwater at STP Upland under industrial and residential land use scenarios are presented in Table 1-1 and were evaluated in the site-specific risk evaluation.

2.3 Estimation of Contaminant Intakes

This step involved estimating EPCs from the analytical data in each media at STP Upland, followed by estimation of the contaminant intake from the EPCs. The detailed description of the methods used to estimate EPCs is presented in Section 1.4.9. As noted, the lower value of the maximum concentration and the 95-percent UCL on the mean concentration was used to estimate potential exposures to the receptors. Tables 2-6 and 2-7 present EPCs used for intake calculation.

2.4 Risk Characterization

Potential exposures to onsite soil (0- to 2-foot interval) and groundwater via the exposure pathways identified in Section 1.4.8 were quantitatively evaluated for each exposure scenario. The calculated HIs and ELCRs for chemicals (Table 2-8) for each receptor are compared to the USEPA's acceptable risk levels and the results are discussed in this section. For radionuclide exposure, potential risks were estimated for residential, construction workers and maintenance worker scenarios using RESRAD (ANL 2001). The RESRAD methodologies are described in Appendix B. The summary of modeled ELCR results obtained from RESRAD is presented in Table 2-9.

2.4.1 Maintenance Worker

2.4.1.1 Onsite Soil—Ingestion, Dermal Contact, and Inhalation of Chemicals

For the maintenance worker scenario, the ELCR from chemicals in soil is 8×10^{-6}, due principally to arsenic in soil. The noncancer hazard index, aggregated across all critical effects and target organs fell below 1.0. Results are presented in Table 2-8.

2.4.1.2 Radionuclides

For the maintenance worker, the ELCR from radionuclides in soil is 2×10^{-3}, due principally to external exposure to Ra-228 and Th-228 in soil. Results are presented in Table 2-9.

2.4.2 Construction Worker

2.4.2.1 Onsite Soil—Ingestion, Dermal Contact, and Inhalation of Chemicals

For the construction worker scenario, the ELCR from chemical contaminants in soil is 8×10^{-8}. The noncancer hazard index, aggregated across all critical effects and target organs fell below 1.0. Results are presented in Table 2-8.

2.4.2.2 Groundwater—Dermal Contact of Chemicals

For the construction worker scenario, the ELCR from chemical contaminants in groundwater is 1×10^{-8}. The noncancer hazard index, aggregated across all critical effects and target organs fell below 1.0. Results are presented in Table 2-8.

2.4.2.3 Radionuclides

For the construction worker scenario, the ELCR from radionuclides in soil is 2×10^{-5}, due principally to external exposure to Ra-228 and Th-228 in soil. Results are presented in Table 2-9.

2.4.3 Residential

2.4.3.1 Onsite Soil—Ingestion, Dermal Contact, Inhalation and Fruit and Vegetable Ingestion of Chemicals

For the residential scenario, the ELCR is 9×10^{-5} from chemical contaminants in soil, with the risk drivers being arsenic and benzo(a)pyrene. The noncancer HI values for iron (noncancer effects in the blood) and arsenic (noncancer effects in the skin) were slightly higher than 1.0. Results are presented in Table 2-8.

2.4.3.2 Radionuclides

For the residential scenario, the ELCR from radionuclides in soil is 4×10^{-3}, due principally to external exposure to Ra-228 and Th-228 in soil. The risk from radon under a residential scenario is 2×10^{-3}. Results are presented in Table 2-9.

2.4.4 Noncancer Risks from Uranium

The noncancer risks from uranium were evaluated in Appendix D, and the results of the evaluation are shown in Table D-3. The noncancer HI for a residential scenario was less than 1.0, indicating that toxic effects from uranium in soil are not a concern.

2.4.5 Risk Characterization for Lead

Lead exceeded the USEPA's recommended lead screening level for residential land use of 400 milligrams per kilograms (mg/kg) (USEPA 1994) in two of the 161 samples collected from STP Upland site. However, the calculated lead EPC of 69 mg/kg (Table 2-6) is less than the screening level, indicating the exposure to lead in soil is not expected to be a concern.

2.5 Uncertainties

The maximum detected concentrations of PAHs in soil were used as the EPCs due to small samples size (< 10 samples). The use of the maximum detected concentrations as EPCs assumes that receptors are exposed daily to these concentrations. Since concentrations of these chemicals in soil are lower at other locations, use of the maximum detected concentrations to represent lifetime average concentrations in soil may overestimate potential exposures, and, consequently associated health risks from PAHs in soil.

2.6 Conclusions

The contaminants providing the largest overall contribution to total site risks at STP Upland were radionuclides in soil. Risks from radionuclides in soil were higher than USEPA's target risk range for the residential and maintenance worker exposure scenarios. Risks from chemical contaminants generally fell within the target risk range, except for iron and arsenic under a residential scenario. In that case, noncancer risks were slightly higher than a target hazard index of 1.0.

SECTION 3

Summary of HHRA Results for the STP River

This section presents a summary of HHRA results conducted on the STP River using methodologies presented in Section 1.

3.1 Data Evaluation and Chemicals of Potential Concern

This risk assessment was performed for the STP River using analytical data obtained during the RI field activities performed from 1993 through 1995 and subsequent investigation conducted in 1999 through 2001. The list of samples used for HHRA is presented in Table 3-1 and detailed description of samples and data collection activities can be found in Section 3 of the Kerr-McGee RI report for the Kress Creek and STP Sites (BBL 2004).

3.1.1 Sediment/Floodplain Soil

Fifty-nine surface sediment/floodplain soil samples (0- to 2-foot interval) were collected from the STP River and were analyzed for metals, VOCs, SVOCs, and radionuclides. Table 3-2 summarizes analytical results of detected contaminants in STP River sediment/floodplain soil. Two metals (arsenic and iron) and one SVOC (benzo(a)pyrene) were detected at concentrations higher than screening levels in sediment/floodplain and were identified as COPCs. Additionally, five radionuclides (radium-226, radium-228, thorium-232, uranium-235, and uranium-238) were identified as COPCs surface sediment/floodplain soil. Table 3-3 presents a summary of COPC selection process for the STP River sediment/floodplain soil.

Two subsurface sediment/floodplain soil samples were collected from the depth interval 0 to greater than 2 feet below surface and were analyzed only for radionuclides. Four radionuclides (radium-226, radium-228, thorium-232, and uranium-238) were identified as COPCs for subsurface sediment/floodplain soil.

3.1.2 Surface Water

Eighteen surface water samples were collected from the STP River and were analyzed for metals and radionuclides. Table 3-4 summarizes analytical results of detected contaminants in STP River surface water. Five metals (antimony, arsenic, iron, manganese, and thallium) were detected at concentrations higher than screening levels in surface water and were identified as COPCs. Additionally, five radionuclides (radium-226, radium-228, thorium-232, uranium-235, and uranium-238) were identified as COPCs for surface water. Table 3-5 presents a summary of COPC selection process for the STP River surface water.

3.1.3 Fish Tissue

Eighteen fish samples were collected from the STP River and were analyzed for metals and radionuclides. Table 3-6 summarizes analytical results of detected contaminants in STP River fish. Eleven metals (aluminum, cadmium, total-chromium, copper, cobalt, iron, lead, manganese, nickel, thallium, and zinc) were detected at concentrations higher than

screening levels in fish tissues and were identified as COPCs. Additionally, thorium-232 was identified as a COPC in fish. Table 3-7 presents a summary of COPC selection process for the STP River fish.

3.2 Potential Exposure Pathways

Potential exposure pathways from STP River sediment/floodplain soil, surface water, and fish consumption under residential and recreational scenarios are presented in Table 1-1 and were evaluated in the site-specific risk evaluation.

3.3 Estimation of Contaminant Intakes

This step involved estimating EPCs from analytical data in each media at STP River, followed by estimation of the contaminant intake from the EPCs. The detailed description of the methods used to estimate EPCs is presented in Section 1.4.9. As noted, the lower value of the maximum concentration and the 95-percent UCL on the mean concentration was used to estimate potential exposures to the receptor. Tables 3-8 through 3-10 present EPCs used for intake calculation.

3.4 Risk Characterization

Potential exposures to sediment/floodplain soil, surface water, and fish via the exposure pathways identified in Section 1.4.8 were quantitatively evaluated for residential and recreational exposure scenarios. The calculated cumulative HIs and ELCRs (across all chemicals and media) for each receptor (Table 3-11) are compared to the USEPA's acceptable risk levels, and the results are discussed in this section. For radionuclide exposure, potential risks were estimated for residential and recreational scenarios using RESRAD (ANL 2001). The RESRAD methodologies are described in Appendix B. Table 3-12 summarizes the modeled ELCR results obtained from RESRAD.

3.4.1 Chemicals

3.4.1.1 Sediment/Floodplain Soil—Ingestion, Dermal Contact, Inhalation and Fruit and Vegetable Ingestion

For the residential scenario, the ELCR is 6×10^{-5} from the chemical contaminants in soil with the risk drivers being arsenic and benzo(a)pyrene. The noncancer HI value for iron (noncancer effects in the blood) was slightly higher than 1.0.

For the recreational scenario, the ELCR is 5×10^{-6}, with the risk driver being arsenic. The noncancer HI value aggregated across all critical effects and target organs fell below 1.0. Results are presented in Table 3-11.

3.4.1.2 Surface Water—Ingestion and Dermal Contact

For the recreational scenario, the ELCR is 6×10^{-7} from chemical contaminants in surface water. The noncancer hazard index, aggregated across all critical effects and target organs, fell below 1.0.

For the residential scenario, the ELCR is 4×10^{-6}. The noncancer HI value, aggregated across all critical effects and target organs, fell below 1.0. Results are presented in Table 3-11.

3.4.1.3 Fish—Ingestion

For the residential and recreational scenarios, the noncancer hazard index, aggregated across all critical effects and target organs, fell below 1.0 for exposure to metals from ingestion of fish. Carcinogenic chemicals were not detected in fish tissue, so ELCR was not evaluated. Results are presented in Table 3-11.

3.4.2 Radionuclides

3.4.2.1 Surface Sediment/Floodplain Soil

For the residential scenario, the ELCR from radionuclides in soil is 4×10^{-3} due principally to external exposure to Ra-228 and Th-228 in soil (see Table 3-12). The risk from radon under a residential scenario is 2×10^{-3}.

For the recreational scenario, the ELCR from radionuclides in soil is 6×10^{-4} due principally to external exposure to Ra-228 and Th-228 in soil (see Table 3-12).

3.4.2.2 Fish Tissue

The ELCR associated with ingestion of radionuclides in fish tissue (under a residential exposure scenario) is 3×10^{-5} (see Table 3-13).

3.4.2.3 Surface Water

The ELCR from the incidental ingestion of radionuclides in surface water is 7×10^{-7} under a residential exposure scenario, and 1×10^{-7} under a recreational scenario (See Table 3-14).

3.4.3 Noncancer Risks from Uranium

The noncancer risks from uranium were evaluated in Appendix D. The results of the evaluation are shown in Table D-3. The noncancer HI for residential scenarios was less than 1.0, indicating that toxic effects from uranium in soil are not a concern.

3.5 Uncertainties

3.5.1 Risks Associated with Chemicals in Sediment/Floodplain Soil

The maximum detected concentration was used as the EPC for benzo(a)pyrene due to the small sample size (< 10 samples). The use of the maximum detected concentration as EPC assumes that receptors are exposed daily to the concentration. Since concentrations of benzo(a)pyrene in sediment/floodplain soil are lower at other locations, use of the maximum detected concentration to represent lifetime average concentration in sediment/floodplain soil may greatly overestimate potential exposures and, consequently, associated health risks for this chemical.

3.5.2 Risks Associated with Chemicals in Surface Water

The maximum detected concentrations were used as the EPCs for metals and radionuclides due to the small sample size (< 10 samples). The use of the maximum detected concentrations as EPCs assumes that receptors are exposed daily to these concentrations. Since concentrations of these contaminants in surface water are lower at other locations, use of the maximum detected concentrations to represent lifetime average concentrations in surface water overestimates potential exposures and, consequently, associated health risks.

3.5.3 Risks Associated with Chemicals in Fish Tissue

The maximum detected concentrations were used as the EPCs for metals and radionuclides due to the small sample size (< 10 samples). The use of the maximum detected concentrations as EPCs assumes that receptors are exposed daily to these concentrations. Since concentrations of these contaminants in fish tissue are lower at other locations, use of the maximum detected concentrations to represent lifetime average concentrations in fish overestimates potential exposures and, consequently, associated health risks.

3.6 Conclusion

The contaminants providing the largest overall contribution to total site risks were radionuclides in soil/sediments. Risks from radionuclides in soil/sediments were higher than USEPA's target risk range for the residential and recreational scenarios. Risks from fish ingestion fell within the USEPA's target risk range, both for chemical and radionuclide contaminants. Risks from chemical contaminants generally fell within the target risk range, except for iron under a residential scenario. In that case, noncancer risks were slightly higher than a target hazard index of 1.0.

SECTION 4

Summary of HHRA Results for Kress Creek

This section presents a summary of HHRA results for the risk assessment conducted at the KCK site using methodologies presented in Section 1.

4.1 Data Evaluation and Chemicals of Potential Concern

This risk assessment was performed for the KCK site using analytical data obtained during the RI field activities performed from 1993 through 1995 and subsequent investigation conducted in 1999 through 2001. The list of samples used for HHRA is presented in Table 4-1 and detailed description of samples and data collection activities can be found in Section 3 of the Kerr-McGee RI Report for the Kress Creek and STP Sites (BBL 2004).

4.1.1 Sediment/Floodplain Soil

One hundred and fifty-six surface sediment/floodplain soil samples (0- to 2-foot interval) were collected from the KCK site and were analyzed for metals, VOCs, SVOCs, pesticides/ polychlorinated biphenyls (PCBs), and radionuclides. Table 4-2 summarizes analytical results of detected contaminants in KCK sediment/floodplain soil. Four metals (arsenic, iron, lead, and manganese), one pesticide (PCB-1260), and four SVOCs (benzo(a)anthracene, benzo(a)pyrene, benzo(b)fluoranthene, and indeno(1,2,3-c,d)pyrene) were detected at concentrations higher than screening levels in sediment/floodplain and were identified as COPCs. Additionally, five radionuclides (radium-226, radium-228, thorium-232, uranium-235, and uranium-238) were identified as COPCs in surface sediment/floodplain soil. Table 4-3 summarizes the COPC selection process for the KCK sediment/floodplain soil.

Thirteen subsurface sediment/floodplain soil samples were collected from the depth interval greater than 2 feet below surface and were analyzed only for radionuclides. Four radionuclides (radium-226, radium-228, thorium-232, and uranium-238) were identified as COPCs in subsurface sediment/floodplain soil.

4.1.2 Surface Water

Twelve surface water samples were collected along KCK and were analyzed for metals. Table 4-4 summarizes analytical results of detected chemicals in KCK surface water. One metal (manganese) was detected at concentrations higher than screening levels in surface water and was identified as a COPC. Table 4-5 presents a summary of the COPC selection process for the KCK surface water.

4.1.3 Fish Tissue

Twenty-seven fish samples were collected from the KCK site and were analyzed for metals and radionuclides. Table 4-6 summarizes analytical results of detected contaminants in KCK fish. Seven metals (arsenic, total-chromium, lead, mercury, nickel, selenium, and thallium) were detected at concentrations higher than screening levels and were identified as COPCs.

Additionally, three radionuclides (radium-228, thorium-232, and uranium-238) were identified as COPCs for fish. Table 4-7 summarizes the COPC selection process for the KCK fish.

4.2 Potential Exposure Pathways

Potential exposure pathways from KCK sediment/floodplain soil, surface water, and fish consumption under residential and recreational scenarios are presented in Table 1-1 and were evaluated in the site-specific risk evaluation.

4.3 Estimation of Chemical Intakes

This step involved estimating EPCs from analytical data in each media at KCK, followed by estimation of the contaminant intake from the EPCs. The detailed descriptions of the methods used to estimate EPCs are given in Section 1.4.9. As noted, the lower value of the maximum concentration and the 95-percent UCL on the mean concentration were used to estimate potential exposures to the receptor. Tables 4-8 through 4-10 present EPCs used for intake calculations.

4.4 Risk Characterization

Potential exposures to sediment/floodplain soil, surface water, and fish via the exposure pathways identified in Section 1.4.8 were quantitatively evaluated for residential and recreational exposure scenarios. The calculated cumulative HIs and ELCRs (across all chemicals and media) for each receptor (Table 4-11) are compared to the USEPA's acceptable risk levels, and the results are discussed in this section. For radionuclide exposure, potential risks were estimated for residential and recreational scenarios using RESRAD (ANL 2001). The RESRAD methodologies are described in Appendix B. Table 4-12 summarizes the modeled ELCR results obtained from RESRAD.

4.4.1 Chemicals

4.4.1.1 Sediment/Floodplain Soil—Ingestion, Dermal Contact, Inhalation and Fruit and Vegetable Ingestion

For the residential scenario, the ELCR is 1×10^{-4} from the chemical contaminants in soil, with the risk drivers being arsenic, PCB-1260, benzo(a)anthracene, benzo(a)pyrene, and benzo(b)fluoranthene). The noncancer HI value for arsenic (noncancer effects in the skin) was slightly higher than 1.0. For the recreational scenario, the ELCR is 1×10^{-5} with the risk drivers being arsenic and benzo(a)pyrene. The noncancer HI value is less than 1.0. The results are presented in Table 4-11.

4.4.1.2 Surface Water—Ingestion and Dermal Contact

For the residential and recreational scenarios, the noncancer hazard index, aggregated across all critical effects and target organs, fell below 1.0 for chemical contaminants detected in surface water. Carcinogenic chemicals were not detected in surface water, so ELCR was not evaluated. Results are presented in Table 4-11.

4.4.1.3 Fish—Ingestion

For the residential scenario, the ELCR from fish ingestion is 4×10^{-5} with the risk driver being arsenic. The noncancer hazard index is 0.2 from arsenic. For the recreational scenario, the ELCR from fish ingestion is 4×10^{-6} with the risk driver being arsenic. The noncancer hazard index aggregated across critical effects and target organs, fell below 1.0. Results are presented in Table 4-11.

4.4.2 Radionuclides

4.4.2.1 Surface Sediment/Floodplain Soil

For the residential scenario, the ELCR from radionuclides in soil is 2×10^{-2} due principally to external exposure to Ra-228 and Th-228 in soil (see Table 4-12). The risk from radon under a residential scenario is 5×10^{-3}.

For the recreational scenario, the ELCR from radionuclides in soil is 2×10^{-3} due principally to external exposure to Ra-228 and Th-228 in soil (see Table 4-12).

4.4.2.2 Fish Tissue

The ELCR associated with ingestion of radionuclides in fish tissue (under a residential exposure scenario) is 3×10^{-5} (see Table 4-13).

4.4.3 Noncancer Risks from Uranium

The noncancer risks from uranium were evaluated in Appendix D, and the results of the evaluations are shown in Table D-3. The noncancer HI for a residential scenario was less than 1.0, indicating that toxic effects from uranium in soil are not a concern.

4.4.4 Risk Characterization for Lead

Lead exceeded the USEPA's recommended lead screening level for residential land use of 400 mg/kg (USEPA 1994) in two of the 93 soil/sediment samples collected from the KCK site for metals analysis. However, the calculated lead EPC of 140 mg/kg (Table 4-8) is less than the screening level, indicating the exposure to lead in soil is not expected to be a concern.

4.5 Uncertainties

4.5.1 Risk Associated with Chemicals in Sediment/Floodplain Soil

The maximum detected concentrations were used as the EPCs for PAHs (benzo(a)anthracene, benzo(a)pyrene, and benzo(b)fluoranthene) and PCB-1260 due to the small samples size (< 10 samples). The use of the maximum detected concentrations as EPCs assumes that receptors are exposed daily to these concentrations. Since concentrations of these chemicals in sediment/floodplain soil are lower at other locations, use of the maximum detected concentrations to represent lifetime average concentrations in sediment/floodplain soil overestimates potential exposures and, consequently, associated health risks.

4.5.2 Risk Associated with Chemicals in Fish Tissue

Calculated ELCR associated with ingestion of fish is 4×10^{-5} with arsenic, which may not be associated with site activities, as the main contributor. Arsenic was detected in one out of the 15 fish samples. Therefore, a concentration equal to one-half of the reporting limit was used instead of reporting zero concentration for samples in which arsenic was not detected. This may result in an overestimate of potential assumed risk associated with arsenic in fish tissue. If the chemical really was present at a concentration of one-half the reporting limit, then typically the laboratory would have reported it as an estimated concentration instead of a non-detect. Additionally, in accordance with USEPA Region VI interim guidance, 30 percent of the arsenic concentrations detected in fish tissue are assumed to be in the carcinogenic form (USEPA Region VI 2002). The calculation of potential risks through ingestion of fish by assuming 100 percent of the arsenic concentration detected in fish tissue is in the carcinogenic form may overestimate the actual risk associated with ingestion of fish.

4.6 Conclusions

The contaminants providing the largest overall contribution to total site risks were radionuclides in soil. Risks from radionuclides in soil were higher than USEPA's target risk range for the residential and recreational scenarios. Risks from fish ingestion fell within the USEPA's target risk range, both for chemical and radionuclide contaminants. Risks from chemical contaminants were at the high end of the target risk range.

SECTION 5

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